External nitrification in biological nutrient removal activated sludge systems

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Abstract

A biological nutrient removal (BNR) activated sludge (AS) scheme incorporating external nitrification in a fixed media system is proposed. A laboratory-scale evaluation of the scheme indicates that it holds considerable potential for BNRAS system intensification through major reduction in sludge age and oxygen demand and significant improvement in sludge settleability. Because the BNRAS system is not required to nitrify, its anoxic mass fraction can be considerably enlarged at the expense of the aerobic mass fraction creating conditions that allow it to achieve high N removals with domestic wastewaters with high TKN/COD ratios; and promote anoxic P uptake polyphosphate accumulating organisms (PAO) to develop in the system. From this, and earlier investigations with conventional nitrification denitrification biological excess P removal (NDBEPR) systems, it appears that anoxic P uptake BEPR is only about $^{2}/_{3}$ of aerobic P uptake BEPR. Hence, inclusion of aerobic P uptake PAOs in the BNRAS system is desirable for the proposed scheme. However, conditions that promote aerobic P uptake to maximise BEPR, are also conducive to nitrifier growth, which, if supported in the BNRAS system, would require virtual complete nitrification in the fixed media system to avoid nitrate interference with BEPR. Before the scheme can be implemented at large scale, an engineering and economic evaluation is required to quantify its potential benefits and savings.

Introduction: The long sludge age requirement for nitrification

The requirement to nitrify governs the sludge age of the biological nutrient removal activated sludge (BNRAS) system. For maximum specific growth rates of nitrifiers at 20°C (μ_{nm20}) around 0.45/d, to guarantee nitrification, the sludge age of the single sludge system must be around 20 to 25 d at 14°C, if 40 to 50% of the sludge mass in the system is aerated. Such long sludge ages result in large biological reactors per Mℓ wastewater (WW) treated. To reduce the sludge age, and hence the biological reactor volume per Mℓ WW treated, internal fixed media such as Ringlace™ have been placed in the aerobic reactor (Wanner et al., 1988; Sen et al., 1994, 1995; Randall and Sen, 1996). The nitrifiers grow on the fixed media establishing a population permanently resident in the aerobic reactor. These nitrifiers are not subject to either the aerobic sludge mass fraction or the suspended mixed liquor sludge age, with the result that both can be reduced. Such a reduction in system sludge age is particularly beneficial for low temperature WWs (10 to 15°C). However, the effectiveness of the internal fixed media has not been as good as expected, and yields a rather low cost/benefit ratio.

It is proposed that external nitrification, i.e. external to the BNRAS system, will provide a more effective reduction in sludge age and aerobic mass fraction. If nitrification can be achieved independently of the BNRAS mixed liquor, the sludge age can be reduced from the usual 20 to 25 d to less than half, around 8 to 10 d. The reduction in sludge age increases the WW treatment capacity of the system by some 50% or, alternatively, reduces the biological reactor volume requirement per Mℓ WW treated by about a $^{1}/_{3}$, without negatively impacting either biological N or P removal: In fact, a reduction in sludge age increases both biological N and P removal per mass organic load (WRC, 1984; Wentzel et al., 1990) and this would be particularly beneficial for low temperature WWs (10 to 15°C). Because nitrification is no longer required, the aerobic mass fraction is governed by the P uptake process, for which aerobic mass fractions can be smaller than for nitrification.

Implementation of external nitrification

External nitrification can be achieved at wastewater treatment plants (WWTPs) where old trickling filter (TF) plants have been extended with a BNRAS system. There are many such WWTPs, particularly in South Africa. Often at these WWTPs, to retain the benefit of the old TF, a proportion of influent WW is passed through the TF and the effluent is (see Fig. 1):

- Discharged to the BNRAS system for biological N and P removal (e.g. Van Huyssteen et al., 1990). This in effect increases the TKN/COD and P/COD ratios of the WW discharged to the BNRAS system and increases the effluent N and P concentrations.
- Chemically treated to precipitate the P before discharge to the BNRAS system. This is not only costly, but also reduces the alkalinity of the water and only reduces the effective P/COD ratio of the WW on the BNRAS system.
- Irrigated on land at the WWTP. This practice will soon be carefully scrutinised in South Africa because it leads to a significant loss of valuable surface water.

If, instead of the above three strategies, the nitrification process is transferred to the TF, all the WW flow can be discharged to the BNRAS system (Fig. 2): A side-stream of mixed liquor is taken from the end of the anaerobic zone and passed through the TF 'humus' tanks (upgraded to internal secondary settling tanks) to remove the AS solids. The underflow sludge is discharged to the beginning of the anoxic zone and the overflow is passed onto the TF for nitrification. The nitrified TF effluent is then discharged to

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Received 12 December 1998; accepted in revised form 21 January 2000.

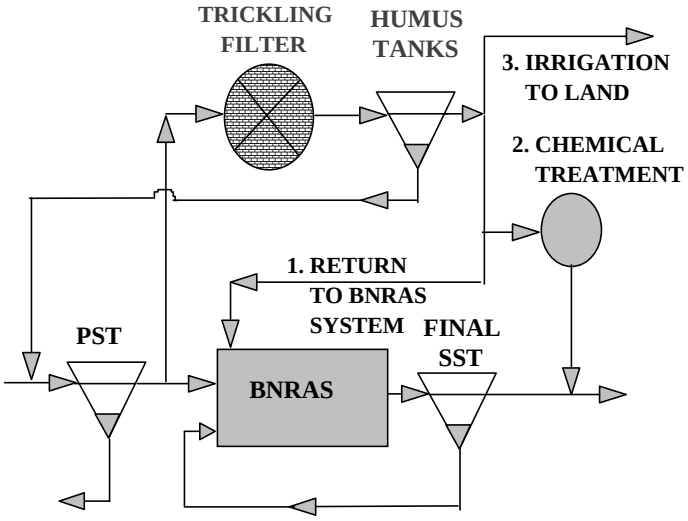

Figure 1
Conventional integration of trickling filters with biological nutrient removal activated sludge systems

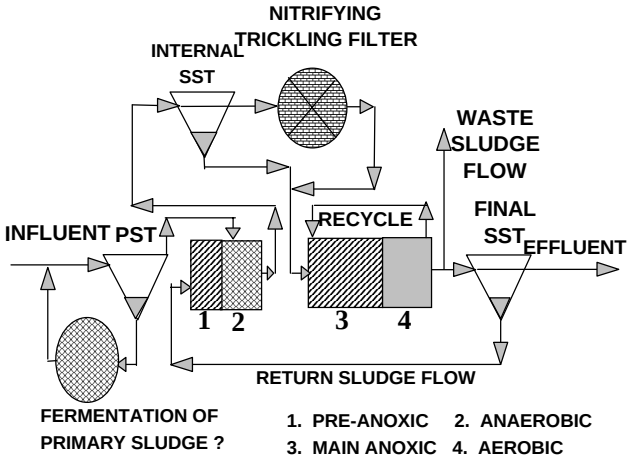

Figure 2
Proposed integration of trickling filters with biological nutrient removal activated sludge systems: nitrification is achieved externally on nitrifying trickling filters

the anoxic zone for denitrification. In this way the TF assists the BNRAS system in its area of weakness, i.e. nitrification, rather the taking away from its strength, i.e. biological N and P removal with influent organics. Furthermore, the oxygen demand in the aerobic reactor is markedly reduced because nitrification no longer takes place there. Indeed, not only is the nitrification oxygen demand obtained "free" outside the BNRAS system, but also the oxygen equivalent of the nitrate generated in the trickling filter helps to reduce the carbonaceous oxygen demand in the BNRAS system, by about a $^1/_3$. In fact, with external nitrification, the reduction in oxygen demand in the BNRAS system is much greater than when $^1/_3$ of the WW is bypassed to the trickling filter as in existing TF/BNRAS systems (Fig. 1). Therefore, by changing the TF to a nitrifying system as in Fig. 2, the treatment capacity of the BNRAS plant is increased without having to increase aeration capacity. If a TF plant is not available, it is possible to include artificial fixed media systems, the cost of which may be offset by the increase in WW treatment capacity.

At short sludge ages and small aerobic mass fractions, nitrifiers would not ordinarily be supported in the BNRAS system. However, it will not be possible to completely exclude nitrifiers from the system because nitrifiers are likely to be seeded into the system from the TF effluent. Therefore, the potential for nitrification in the aerobic reactor will always exist in the system, and the potential nitrate concentration in the aerobic reactor will be governed by the ammonia concentration that enters it. Provided the TF nitrifies well, this nitrate concentration will be mainly from the ammonia which bypasses the TF via the internal settling tank underflow, and therefore will be relatively low. If the TF does not nitrify well and the residual ammonia concentration from it is high, then, if sufficient nitrifiers are present in the aerobic reactor, the nitrate concentration will be high, with the result that a significant nitrate concentration will be present in the underflow from the final settling tank. To protect the BEPR against this potential nitrate ingress to the anaerobic reactor, a pre-anoxic reactor is placed in the underflow to denitrify the nitrate (Fig. 2). If sufficient nitrifiers are not present in the aerobic reactor, then the ammonia concentration in the aerobic reactor will only be partially nitrified with the result that return sludge nitrate concentration will be relatively low, but the effluent TKN concentration will be high, the concentration depending on the nitrification efficiency of the TF.

Tertiary nitrifying trickling filters (TNTFs), which are employed for nitrification only and negligible organic material removal, are fairly common in the USA (Lutz et al., 1990). While certain problems with macrofauna (snails, worms, larvae and flies), which reduce nitrification rates, have been encountered, high removals of ammonia have been economically achieved in TNTFs (Parker et al., 1989; 1995, 1996). Therefore, while some full-scale trials would need to be done to determine how rock media trickling filters can be adapted to fulfill the external nitrification function, from the USA experience this is not expected to be a major problem in implementing the external nitrification scheme (Fig. 2).

With nitrification taking place externally to the BNRAS system, the aerobic mass fraction of the BNRAS system can be significantly decreased, from the usual 40 to 60% to as low as 15 to 25%. Such low aerobic mass fractions, which allow high anoxic mass fractions, appear to stimulate a shift in the biological excess P removal (BEPR) behaviour. Whereas usually P uptake takes place predominantly under aerobic conditions, with large anoxic mass fractions and a sufficiently high nitrate load, significant (>40%) P uptake can take place under anoxic conditions. This anoxic P uptake has been increasingly reported in the past five years (Kerrn-Jespersen and Henze, 1993; Kuba et al., 1993) and also has been found to take place in conventional nutrient removal activated sludge systems at laboratory scale (Ekama and Wentzel, 1999) and full scale (Kuba et al., 1997). If complete P uptake can be achieved under anoxic conditions, it may be possible to eliminate the aerobic zone altogether, and, from a BEPR process point of view, this has been demonstrated at laboratory scale to be feasible (Kuba et al., 1996). However, from a liquid/solid separation point of view, a small re-aeration reactor will still be required to improve the sludge's clarification characteristics and reduce effluent turbidity before final settling. It would appear therefore, that small aerobic mass fractions may not restrict the functionality of the BEPR process. However, there are indications that anoxic P uptake BEPR is not as high as aerobic P uptake BEPR with respect to the volatile fatty acids (VFA) taken up in the anaerobic zone (Ekama and Wentzel, 1999).

The large anoxic mass fraction (50 to 70%) in the BNRAS system with external nitrification would ordinarily allow complete denitrification to be achieved in the anoxic reactor (depending on

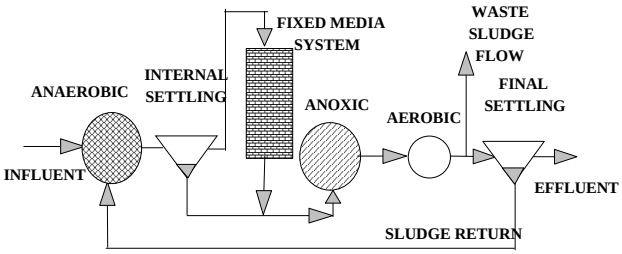

Figure 3
The DEPHANOX biological nutrient removal system (after Bortone et al., 1996: Sorm et al., 1996)

the influent TKN/COD ratio). Complete denitrification in the anoxic reactor preceding the aerobic reactor, together with the short sludge age (8 to 10 d), are conditions that are hypothesised to ameliorate the AA (low F/M) filament bulking problems common in conventional BNR plants (Casey et al., 1994). Low DSVIs (<75 mℓ/g) increases the treatment capacity of the BNRAS system by a further approximately 50% compared with conventional BNR plants, which often have rather poor settling sludges (DSVI >150 mℓ/g) and therefore are designed with large SSTs. Hence external nitrification allows a major step increase in BNRAS system intensification, by approximately doubling the WW treatment capacity or halving the biological reactor volume requirements per Mℓ WW treated. This system intensification may compensate for the cost of the internal settling tank after the anaerobic reactor and the additional pumping costs. However, an engineering and economic evaluation of the external nitrification BNRAS scheme is needed to more accurately quantify the cost benefits of the scheme.

To exploit anoxic P uptake as much as possible, Bortone et al. (1996) developed the DEPHANOX system (Fig. 3). In this system, nitrification also takes place externally to the BNRAS system, to allow large anoxic and small aerobic mass fractions to stimulate anoxic P uptake. This system is similar to the external nitrification scheme proposed in this paper (Fig. 2) except that in the latter, small aerobic mass fractions to exploit anoxic P uptake is not the primary objective and expressly suppressing nitrifier growth in the BNRAS system is not essential for the protection of the BEPR.

Laboratory- and pilot-scale experimental work has been done on the DEPHANOX system by Bortone et al. (1996) and Sorm et al. (1996). They found that in the anoxic reactor, denitrifying PAOs (DPAOs) participate in the denitrification process presumably utilising internally stored polyhydroxyalkanoates (PHA) formed in the anaerobic reactor during volatile fatty acid (VFA) uptake and P release. Therefore, together with the ordinary heterotrophic organisms (OHOs), denitrification by DPAOs with P uptake also takes place in the anoxic reactor. Improved sludge settleability (SVIs 50 mℓ/g) have been consistently observed in a laboratory-scale DEPHANOX system by Sorm et al. (1996).

Objectives of research

From the above discussion, the external nitrification scheme seems to hold considerable promise for increasing BNR WWTP capacity, but it needs to be tested at laboratory scale before it can be implemented at full scale. Also, some important questions need to be answered to check whether or not it can realise the potential to increase WWTP capacity, viz.:

- Does the system consistently produce a good settling sludge?
- If anoxic P uptake BEPR is to be exploited in the system, is anoxic P uptake BEPR as good as aerobic P uptake BEPR?
- What factors promote anoxic P uptake?

In this paper, the results of a laboratory-scale investigation into the external nitrification BNR scheme are presented. The results are evaluated to address at least in part the questions above, and to assess the system performance and its potential for application at a larger scale.

Experimental investigation

Experiment system setup

External nitrification in a laboratory-scale **non-nitrifying** BNRAS system can be achieved in two ways, either by installing a settling tank and biofilm or trickling filter system between the anaerobic and anoxic reactors, as shown in Fig. 2; or dosing nitrate directly to the anoxic reactor. To set up the laboratory system as close as possible to the envisioned application, the first approach was adopted. However, in order to increase the nitrate load on the main anoxic reactor to assess its denitrification potential, nitrate was also dosed directly to the anoxic reactor. Accordingly, the system was set up as shown in Fig. 4, with the design parameters as given in Table 1. The system sludge age, temperature and aerobic sludge mass fraction were 10 d, 20°C and 0.20 respectively. For nitrifiers to be supported in the suspended mixed liquor under these conditions, their maximum specific growth rate at 20°C (μ_{nm20}) would

TABLE 1
EXTERNAL NITRIFICATION BNRAS SYSTEM DESIGN AND OPERATING PARAMETERS

Parameter	Value	Parameter	Value
Operating:		**Reactor volumes (ℓ) and mass fractions (%):**	
Sludge age (d)	10		
Temperature (°C)	20	Pre-anoxic	2[#] ; 9.5%
pH of anaerobic reactor	7.2 - 8.2	Anaerobic	5 ; 23.8%
pH of aerobic reactor	7.2 - 8.2	Main anoxic	10 ; 47.6%
DO in aerobic reactor (mgO/ℓ)	2.0 - 5.0	Aerobic	4 ; 19.0%
		Unaerated mass fraction	17 ; 81.0%
		Stone column	11.8 ℓ;
Influent: Mitchell's Plain raw		**Recycles:** ratio ; ℓ/d	
Flow (ℓ/d)	20	Underflow (s-recycle)	1:1 ; 20
COD (mgCOD/ℓ)	750	To stone column	1.75:1; 35
RBCOD (mgCOD/ℓ)	~110	Underflow (internal SST)	0.25:1; 5
TKN/COD ratio	0.06-0.11		
Total P (mgP/ℓ)	13.6-28.9		

[#]Actual volume is 1 ℓ, but with s = 1:1, the pre-anoxic VSS concentration is double that in the remainder of the system; therefore the equivalent volume at system VSS conc. is 2 ℓ.

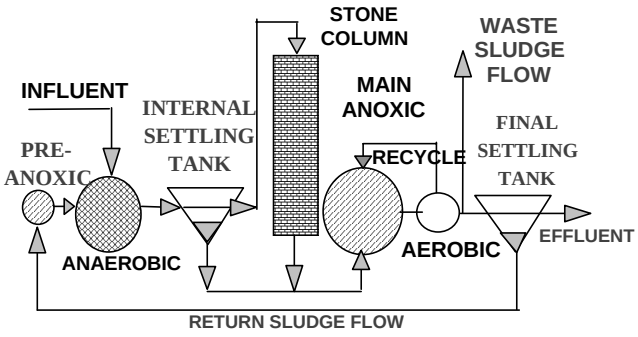

Figure 4
Experimental setup for external nitrification biological nutrient removal activated sludge system

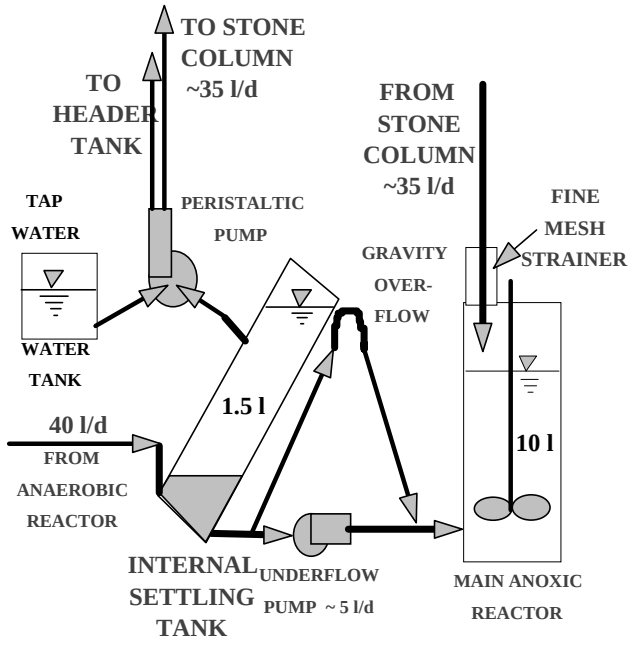

Figure 5
Detail of stone column flow control and measurement

need to be greater than 0.70 /d. This is a high value so growth of nitrifiers in the activated sludge was unlikely. However, to avoid the possible interference of nitrate on the P release process in the anaerobic reactor, a 0.10 mass fraction pre-anoxic reactor was installed ahead of the anaerobic reactor to denitrify nitrate in the return sludge flow. This pre-anoxic reactor would ensure a zero nitrate flow to the anaerobic reactor if nitrate is dosed in excess to the main anoxic reactor resulting in nitrate in the outflow of this reactor; and/or if the residual ammonia is nitrified in the aerobic reactor.

The activated sludge reactors and settling tanks were made from clear acrylic plastic as described in detail by Clayton et al. (1989). The fixed medium reactor was a 100 mm diameter, 1.5 m tall clear acrylic plastic column filled with 4 to 6 mm gravel. The base was closed with a flat flange to retain the stones in the column, but four 5 mm diameter holes around the circumference of the column base allowed outflow of nitrified effluent and inflow of air. A small fan at the top of the stone column operated for 30 s every 30 min to renew the air in the column because it was too small to generate its own natural upflow air draught. The column stood in a 200 mm diameter 100 mm high cylindrical acrylic plastic tank to collect its effluent, which drained by gravity into the anoxic reactor. Before discharge to the anoxic reactor, the stone column effluent was passed through a fine mesh (0.1 mm) strainer to remove solids, in particular worms and larvae which appeared to adversely affect the laboratory-scale activated sludge system. [*This strainer prevented the macrofauna of the stone column trickling filter from entering the activated sludge system. Such a macrofauna separator is unlikely to be required in full scale plants because there are many instances at full-scale where trickling filter effluents are discharged to the activated sludge plant without apparent adverse affect on the activated sludge biomass*].

The flow to the stone column was drawn from the internal settling tank supernatant with a peristaltic pump as shown in Fig. 5. To provide control and monitoring of the flow to the stone column, a parallel channel in the peristaltic pump discharged tap water to a header tank with a hydraulic head equal to that of the stone column. Of the 40 ℓ/d passing through the internal settling tank (20 ℓ/d influent plus a 1:1 underflow recycle ratio), between 30 to 35 ℓ/d was pumped to the stone column. The underflow sludge was pumped to the anoxic reactor at about 5 ℓ/d. A gravity overflow from the settling tank to the anoxic reactor absorbed flow variations in the two pumps (see Fig. 5). The complete system was operated in a temperature controlled laboratory at 20°C.

Experimental system operation

Throughout the 250 d investigation, the system was fed 20 ℓ/d raw sewage obtained from Mitchell's Plain WWTP. The raw sewage was collected in batches approximately every two weeks and, after maceration, stored in stainless steel tanks in a cold room controlled at 4°C. To feed the laboratory system, a sample of sewage was drawn from the storage tanks after thorough mixing and then diluted with tap water from its raw COD of approximately 1 100 mg COD/ℓ to a target COD concentration of 700 mg COD/ℓ. The diluted influent was buffered by addition of sodium hydrogen carbonate ($NaHCO_3$). About 10 mg P/ℓ potassium dihydrogen phosphate (KH_2PO_4) was dosed to the influent to avoid P limitation and ensure that the effluent total phosphorus (TP) concentration from the system remained above 5 mg P/ℓ. After taking a sample, the 20 ℓ diluted influent was placed in the system's feed drum maintained at approximately 8°C and served as influent for a 24 h period. Feeding the wastewater in 24 h batches provided a check that daily all the influent COD was discharged to the system. At the end of a 24 h period, any solids that may have accumulated in the bottom of the feed drum were collected and poured into the anaerobic reactor. When nitrate was dosed, this was done directly into the main anoxic reactor with a dose pump delivering 380 mℓ/d. The dose concentration of the $NaNO_3$ solution was varied from 0.5 to 1.25 mgN/mℓ depending on the nitrate mass dose required per day.

System performance monitoring and analytical methods

To monitor the system performance, samples were drawn virtually daily for analysis from each of the reactors, the internal settling tank supernatant, the stone column outflow and final effluent; Table 2 shows the parameters that were measured and the analytical methods applied. Also, the diluted sludge volume index (DSVI) was measured on sludge drawn from the aerobic reactor and the oxygen utilisation rate (OUR) was monitored continuously online with the DO controller/OUR meter developed by Randall et al. (1991). In addition, anaerobic, anoxic and aerobic batch tests were conducted occasionally on sludge drawn from the system to examine the P release, P uptake and denitrification kinetics but these results are not presented in this paper.

TABLE 2
SAMPLING POSITION AND PARAMETER MEASUREMENT

Test	COD[1]	TKN[2]	FSA[3]	NO_3[4]	NO_2[4]	Tot P[5]	OUR[6]	DSVI[7]	V/TSS[8]	pH[9]
Pre-anoxic				†	†	†			✓	
Influent	★†[10]	★	†			★				
Anaerobic				†	†	†			✓	
SC influent	★†	★	†	†	†	†				
SC effluent	★†	★	†	†	†	†				
Anoxic				†	†	†			✓	
Aerobic	★	★		†	†	†	✓	✓	✓	✓
Final effluent	★†[10]	★†	†	†	†	★†				

✓Measurement taken (filtering not applicable); ★ Unfiltered sample; SC = Stone column;
†Filtered through Schleicher & Schüll 0.45 µm glass fibre membrane.
[1,2,3] Method according to *Standard Methods* (1985).
[4]According to Technicon AutoAnalyser Industrial Method No 33.69W.
[5]Sulphuric acid/persulphate digestion at 100°C followed by molybdate-vanadate colour development for orthophosphate (*Standard Methods*, 1985 - Method 424C III).
[6] With Yellow Springs DO probe and the automated method of Randall et al. (1991).
[7] According to Lee et al. (1983) or Ekama and Marais (1984).
[8] By separation with centrifugation, drying in a crucible 105°C and incineration at 600°C.
[9] With Hanna Instruments pH meter No HI 9023.
[10] For the influent readily biodegradable (RB)COD concentration, the influent and effluent samples were subjected to an alum flocculation step prior to filtration (Mamais et al., 1993; Mbewe et al., 1998).

Experimental results

COD and nitrogen mass balance

The system operating and sample analytical procedures and the accuracy of the experimental data were checked by means of COD and N mass balances calculated from the sewage batch averages of the measured parameters. The sewage batch periods were chosen to calculate the averages because the influent TKN and RBCOD concentrations varied from sewage batch to sewage batch. Also, during some sewage batches, nitrate was dosed to the main anoxic reactor to ensure a sufficiently high nitrate load to exceed this reactor's denitrification potential. The sewage batch periods cannot be called "steady state" periods insofar as BEPR is concerned because the anoxic P uptake biomass emerged slowly over time. Indeed, after 250 d, having fed 13 batches of sewage, the P uptake in the anoxic reactor was still increasing while that in the aerobic reactor was still decreasing.

In the COD balance, the COD mass leaving the system via the final effluent flow; oxygen utilised; sludge wasted; nitrate and nitrite denitrified; and COD utilised in the stone column is reconciled with the COD mass entering the system with the influent flow. In the N balance, the N mass leaving the system via the final effluent flow; sludge wasted; nitrate and nitrite denitrified; and nitrogen removed in the stone column is reconciled with the TKN mass entering the system with the influent flow and the nitrate mass dosed. The N nitrified and denitrified is calculated from a nitrate and nitrite mass balance around all the system's reactors, settling tanks and stone column. The reliability of the experimental data is directly proportional to the mass balance deviation from the target value of 100%, i.e. the closer the mass balances are to 100%, the more reliable the data.

Initially, the COD concentrations in the inflow and outflow of the stone column were not measured. When it appeared that significant COD removal was taking place in the stone column, measurement of these concentrations was commenced. Results from the sewage batches where these concentrations were measured indicated that the COD removal in the stone column was about 15% of influent COD and so for the sewage batches where these concentrations were not measured, a 15% COD removal was assumed in the COD balance. This prompted an inquiry into the N removal in the stone column. From measurements of the TKN and nitrate concentrations in the inflow and outflow of the stone column, the N removal was around 12% of influent TKN. This N removal was high for biomass growth only so it appeared that denitrification also took place in the stone column. The fan on-time should have been increased to increase the aerobic conditions in the stone column and so minimise the N removal by denitrification, but this was not done; attention was focused on the BNRAS system performance. The requirement of the stone column was to nitrify and after an initial start-up period (Sewage Batches 1 and 2), this it did satisfactorily throughout the investigation.

The COD and N mass balances obtained for each sewage batch are listed in Table 3. Although lower than 100%, these are similar to COD and N balances observed in other investigations with nutrient removal systems, viz. Clayton et al. (1989), 92% and 91%; Kaschula et al. (1993), 84% and 89%; Pilson et al. (1995), 84% and 97%; Mellin et al. (1998), 84% and 82% and Sneyders et al. (1997), 90% and 92% (see Ekama and Wentzel, 1999). Considering that the COD and TKN losses in the stone column were not accurately monitored until later in the investigation and that the COD and N balances are not worse than those for conventional BNRAS systems operated in the past, the data obtained in this investigation can be accepted for evaluating the BNRAS system with external

nitrification. The mean values of the measured parameters for Sewage Batches 1 to 13 are listed in Table 4.

The COD and N balance components for Sewage Batches 1 to 13 are given in Figs. 6 and 7 respectively. The total height of the stacked bars give the COD and N balance achieved. For the COD balance (Fig. 6), about 8% of the influent COD mass leaves the system via the effluent flow, about 30% via sludge wastage, 16% via nitrate denitrified, 19% via oxygen utilised and 15% in the stone column, leaving 10% unaccounted for. These results indicate that almost as much COD was utilised with nitrate as electron acceptor (16%) as with oxygen as electron acceptor (19%). Nitrate utilisation was higher than would be usual due to the dosing of nitrate to the main anoxic reactor. During Sewage Batches 9 to 13, 20 mgNO_3-N/ℓ influent was dosed, which was about 58% of the nitrate generated in the stone column. The low oxygen demand in the system is reflected in the low OUR in the aerobic reactor, viz. only about 29 mg/(ℓ·h) in 20% of the system volume. The OUR in an equivalent internal nitrification BNRAS system (20% aerobic mass fraction, 90% COD balance, 10 d sludge age, complete nitrification, 90% nitrate denitrification leading to 50% recovery in nitrification OUR) is about 2.5 times higher, i.e. 75 mg/(ℓ·h). Clearly not nitrifying in the BNRAS system and utilising the nitrate generated in the stone column results in a major decrease in OUR. The proportion of COD utilised with nitrate indicates that the denitrification potential of the system is very high, due to the large anoxic mass fraction (50%). This allows the system to deal with very high TKN/COD ratios without jeopardising BEPR, provided near complete nitrification is achieved in the external fixed media system.

For the N balance (Fig. 7), about 12% of the influent N mass leaves the system via the effluent flow. Of this 12%, 7% is TKN and 5% nitrate. About 25% of the influent N leaves the system via sludge wastage, but this includes the N removal in the stone column. Of this 25%, 13% was N removal via sludge wastage from the BNRAS system and 12% in the stone column. A stone column N removal of 12% for sludge production is almost equal to that of the BNRAS system and clearly is too high, indicating that some denitrification took place in the stone column (see above). The N leaving the system via N_2 gas (denitrification) is very high at 52%, of which 48% is denitrification in the main anoxic reactor and 4% in the pre-anoxic reactor. The reason for such a high denitrification N removal is the large anoxic reactor and the dosing of nitrate during the second half of the investigation (Sewage Batches 6 to 13) to realise this reactors full denitrification potential.

TABLE 3
NITROGEN AND COD MASS BALANCES, INFLUENT TKN/COD RATIO, % COD AND % N REMOVALS, SYSTEM P REMOVAL AND NITRATE CONCENTRATION DENITRIFIED IN THE MAIN ANOXIC REACTOR PER LITRE INFLUENT

Sewage batch	N balance %	COD balance %	[1] Influent TKN/COD ratio	%COD removal	%N removal	System P Rem'l mgP/ℓ	Nitrate denit'd mgN/ℓ
1	90.5	84.0	0.121	86.89	69.33	5.8	37.8
2	90.8	82.5	0.120	89.39	67.01	2.2	16.6
3	89.6	89.0	0.082	90.41	87.41	7.0	52.6
4	80.4	81.2	0.109	90.17	80.08	4.6	35.0
5	89.3	80.4	0.094	91.77	88.76	9.1	54.7
6	85.1	90.4	0.109	90.82	90.67	9.2	61.3
7	91.8	88.4	0.099	92.83	91.49	8.4	71.8
8	102.0	90.0	0.094	92.24	93.91	7.6	48.7
9	87.8	94.0	0.142	92.75	90.90	12.6	63.3
10	91.9	94.9	0.121	93.07	91.79	11.9	64.0
11	96.0	98.4	0.108	93.67	91.00	12.5	70.1
12	92.9	92.9	0.121	94.47	89.60	13.9	70.7
13	94.2	91.1	0.137	91.45	83.90	10.2	59.4
Mean	90.9	89.0	0.112	91.5	85.8	9.5[2]	-

[1.] Includes nitrate dose. [2] Sewage batches 4 to 13 inclusive

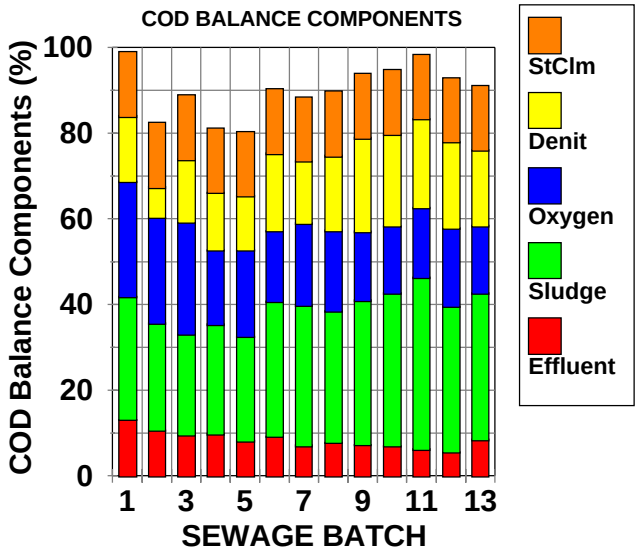

Figure 6
COD balance components showing proportions of influent COD mass leaving system via effluent, sludge wastage, oxygen utilisation, nitrate utilisation and stone column removal

COD removal performance

From Table 3, the system COD removal efficiency was very good at 92%. The unfiltered effluent COD concentration ranged between 43 and 84 with a mean of 60 mgCOD/ℓ. The 0.45 µm membrane filtered effluent COD concentration ranged between 34 and 70 mg COD/ℓ, with a mean of 51 mgCOD/ℓ. This value was accepted to correspond to the unbiodegradable soluble COD in the influent and gives an unbiodegradable soluble COD fraction ($f_{S_{,us}}$) of 0.071. The sewage batch mean unfiltered influent and effluent COD concentrations are given in Fig. 8, which shows that the COD removal performance of the system was not only good, but also stable during the entire investigation.

Nitrogen removal performance

For the 13 sewage batches the N components in the influent and effluent flows are shown in Fig. 9. It can be seen that the influent TKN concentration varied between 60 and 80 mgN/ℓ and the dosed nitrate concentration increased from 9 to 19 mgN/ℓ for Sewage

TABLE 4

AVERAGES OF MEASURED PARAMETERS FOR SEWAGE BATCHES 1 TO 13 AND OVERALL MEAN OF THE 13 SEWAGE BATCHES. THE NO_x GAIN/LOSS, PREL/PUPT AND P REMOVAL DATA, AS WELL AS THE COD AND N BALANCE RESULTS, WERE CALCULATED FROM THESE AVERAGE VALUES.

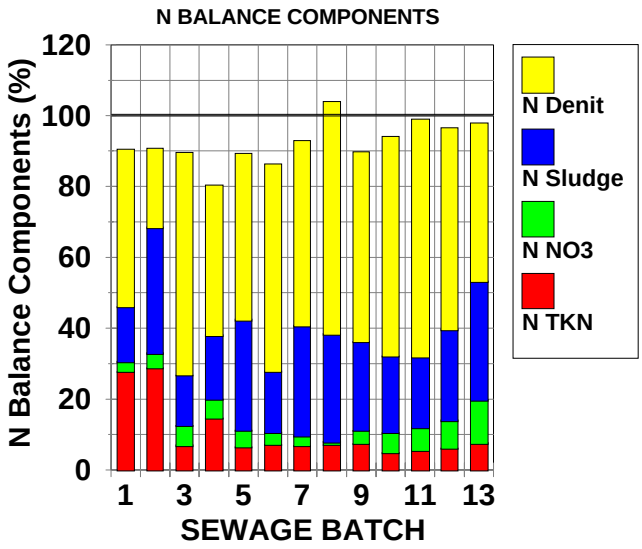

Figure 7
Nitrogen balance components showing proportions of influent TKN and nitrate dosage masses leaving system via effluent TKN and nitrate, sludge wastage (which includes stone N column removal) and denitrification (N_2 gas)

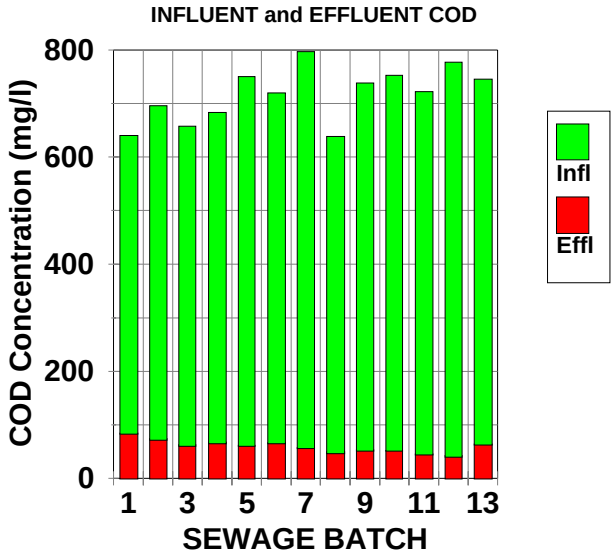

Figure 8
Average sewage batch influent and effluent COD concentrations. Effluent COD concentration is standing in front of the influent COD concentration

Batches 6 to 13, increasing the influent N to between 80 and 100 mgN/ℓ. This yields equivalent influent TKN/COD ratios of 0.11 to 0.14 mgN/mgCOD which can be treated in the system without nitrate recycle to the anaerobic reactor because the stone column nitrified virtually completely. This influent TKN/COD range is high and embraces most well-settled wastewaters and demonstrates that the system is well suited to unfavourable influent TKN/COD ratios while still ensuring good BEPR (i.e. no nitrate recycle into the anaerobic reactor); and a low effluent N concentration. Indeed, the system requires high influent TKN/COD ratios to

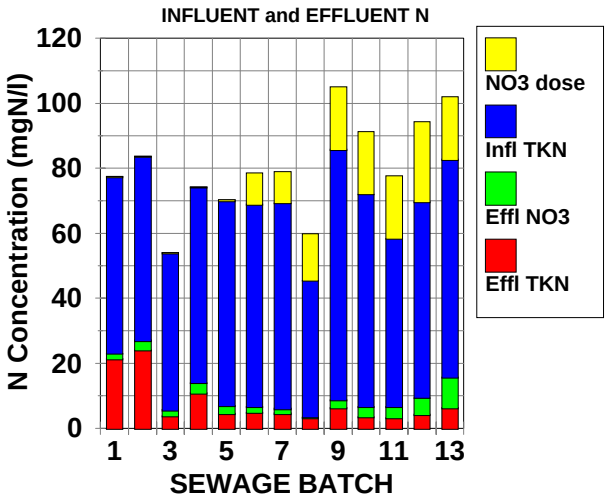

Figure 9
Average sewage batch influent and effluent N concentrations. Effluent TKN and nitrate concentrations are standing in front of the influent TKN and nitrate dosage concentrations.

ensure a sufficiently high nitrate load on the anoxic reactor to stimulate anoxic P uptake (see below). From Fig. 9, the effluent total N was mostly below 10 mgN/ℓ - the mean effluent total N (TKN + NO_x) for Sewage Batches 3 to 13 was 8.8 mgN/ℓ of which 3.7 mgN/ℓ is nitrate and 5.1 is TKN. Only Sewage Batch 13 had a high effluent nitrate concentration (9.4 mgN/ℓ), but this was due to overdosing nitrate to the main anoxic reactor.

In the system, nitrification took place externally in the stone column. The BNRAS effluent free and saline ammonia (FSA) concentration was controlled by the stone column, because very little nitrification took place in the aerobic reactor. This can be seen in Fig. 10, which gives the nitrate/nitrite (NO_x) mass balance results over the pre- and main anoxic, aerobic and stone column reactors of the system. 88% of the nitrate generated was produced in the stone column and only 12% in the aerobic reactor. Also, nitrification was virtually complete; for Sewage Batches 3 to 13, the mean stone column outflow and final effluent FSA concentrations were 4.4 and 3.8 mgN/ℓ respectively. Pumping an average of 35 ℓ/d from a 40 ℓ/d outflow from the anaerobic reactor (88%) to the stone column meant that at best nitrification could be 88% complete if the stone column FSA concentration were zero. On the measured results, nitrification efficiency was 86% (viz. 3.8/27.1) demonstrating that external nitrification should not pose a significant problem in system implementation [*In follow-up experiments on the external nitrification scheme, considerable problems with* Psychoda *larvae and flies were encountered causing a reduced nitrification efficiency (<50%). Because TF nitrification performance cannot be adequately scaled up and therefore would need to be evaluated at full scale, the stone column has been replaced with a satellite rotating biological contactor/activated sludge system*]. Provided the fixed media system nitrifies virtually completely, the proportion of flow pumped to it governs the percentage nitrification, the greater this proportion, the greater the percentage nitrification. Because the sludge in the BNRAS system settled so well (DSVI~50 mℓ/g, see Fig. 14), high internal settling tank underflow sludge concentrations can be achieved and therefore also high proportions of flow (>80%) can be pumped to the fixed media system.

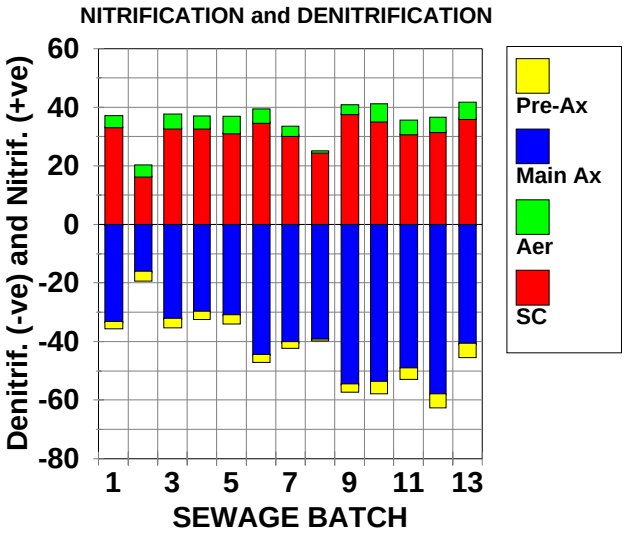

Figure 10

Nitrification in the stone column and aerobic reactor (+ve) and denitrification (-ve) in the main and pre-anoxic reactors in mgN/ℓ influent calculated from nitrate+nitrite mass balance around the reactors and stone column

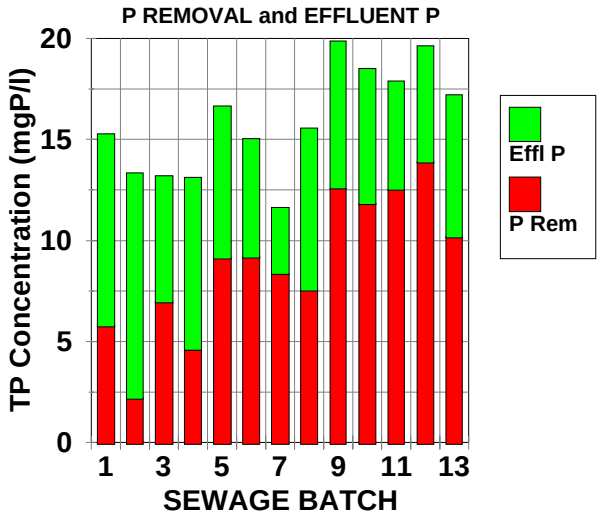

Figure 11

Mean P removal and unfiltered effluent P in mgP/ℓ for the 13 sewage batches. Total height of stacked bar gives influent P concentration.

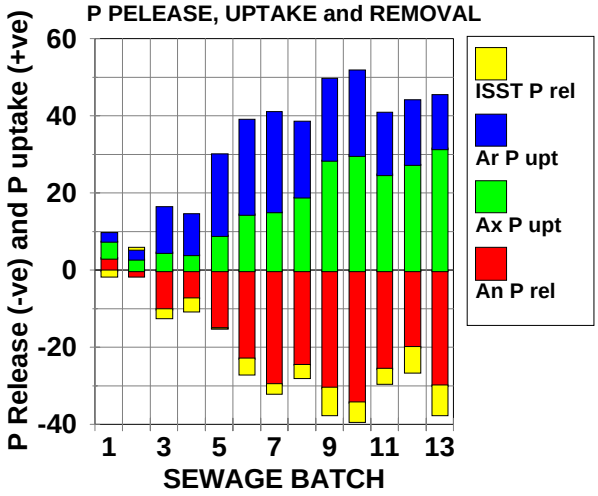

Figure 12

Mean P release (-ve) in the anaerobic and internal settling tank and P uptake (+ve) in the anoxic and aerobic reactors observed during the 13 sewage batches. Total height of stacked bar gives total P release (-ve) and total P uptake (+ve).

Biological excess P removal (BEPR) performance

For the 13 sewage batches, the average sewage batch influent, reactor, settling tank and stone column outflow and effluent TP concentrations are given in Table 4. From these averages, the net P uptake (+ve) or release (-ve) were calculated from a TP balance over each reactor, the two settlers and the stone column of the system. From Table 4, the P uptake and/or P release in the pre-anoxic and final settling tank was negligible.

For each of the 13 sewage batches, the mean influent and effluent P concentrations and system P removal are plotted in Fig. 11. The P release in the anaerobic reactor and internal settling tank and the P uptake in the anoxic and aerobic reactors are plotted in Fig. 12. To highlight the anoxic and aerobic P uptake contributions, these are shown as percentages of the total P uptake in Fig. 13.

From Fig. 11, initially P removal was poor, around 5 mgP/ℓ. This was because the system was started up with non-BEPR sludge from a Modified Ludzack-Ettinger (MLE) N removal system. Over the first 6 sewage batches, the P removal improved to around 9 mgP/ℓ. From Fig. 12, most of the P release took place in anaerobic reactor, but some release continued to take place in the internal settling tank. This is probably due to leakage of some RBCOD out of the anaerobic reactor, stimulating P release in the sludge blanket that formed in the bottom of the internal settling tank due to the low underflow pumping rate to the main anoxic reactor. The P release in the internal settling tank is not undesirable; in fact, in DEPHANOX II (Bortone et al., 1997), the first reactor is a combined anaerobic reactor-settling tank to increase the anaerobic mass fraction per unit volume; and reduce the cost of the internal settling tank in the system.

With regard to P uptake (Fig. 12), initially this was low and mostly in the aerobic reactor. As the investigation continued, the P uptake increased and gradually more took place in the anoxic reactor. For the first 6 sewage batches, the % anoxic P uptake remained approximately constant at about 25% (Fig. 13) and ways to increase this were explored. It was noted from earlier investigations on BEPR in conventional BNRAS systems (see Ekama and Wentzel, 1999 for details), that when the main anoxic reactor was underloaded with nitrate, the P uptake tended to be confined to the aerobic reactor, whereas if the nitrate load exceeded the denitrification potential so that nitrate was present in the anoxic reactor outflow, then significant anoxic P uptake was sometimes observed. Accordingly, nitrate was dosed into the main anoxic reactor and increased stepwise with succeeding sewage batches to avoid nitrate overload on the main anoxic reactor and hence a high nitrate recycle to the anaerobic reactor. Nitrate dosing was commenced with Sewage Batch 6 at 9.63 mgNO_3-N/ℓ influent (see Table 4) and this was maintained for Sewage Batch 7 also. Because the NO_x concentration in the main anoxic reactor continued to be low, the nitrate dose was increased to 14.44 mgNO_3-N/ℓ influent for Sewage Batches 7 and 8. This increased the % anoxic P uptake to around 35% (Fig. 13). For Sewage Batches 9 to 11 the nitrate dose

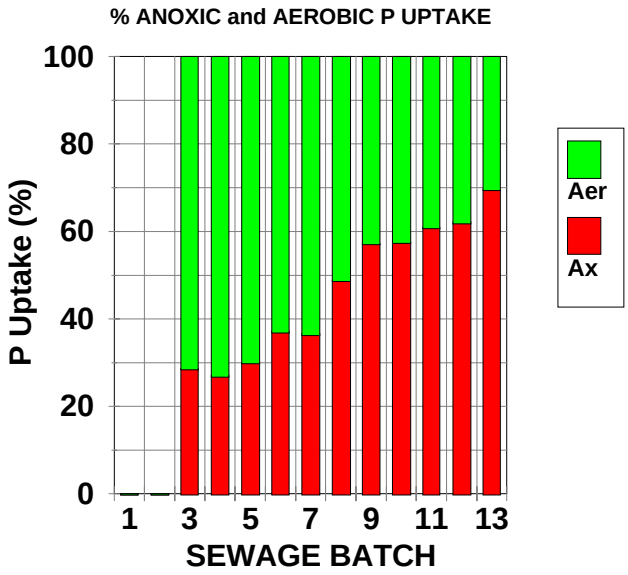

Figure 13
Change in mean percentage anoxic and aerobic P uptake for Sewage Batches 1 to 13

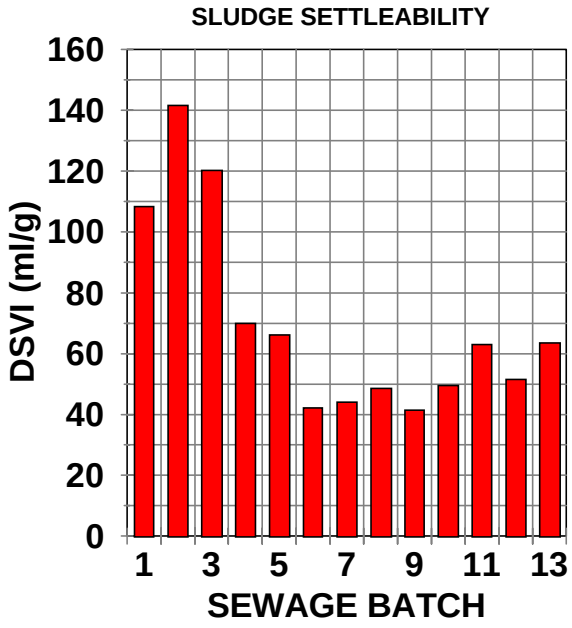

Figure 14
Mean DSVI values measured during Sewage Batches 1 to 13

was increased to 19.25 mgNO_3-N/ℓ influent and the % anoxic P uptake increased further, to around 60%. The main anoxic reactor NO_x concentration was still below 1 mgN/ℓ which indicated that the nitrate load was still below the denitrification potential even though it was denitrifying between 49 and 55 mgNO_3-N/ℓ influent (see Table 4). The P removal also appeared to have stabilised at around 12.5 mgP/ℓ . For Sewage Batch 12, the nitrate dose was increased to 24.6 mgNO_3-N/ℓ influent. The P removal increased to 13.5 mgP/ℓ, but the % anoxic P uptake remained at about 60%. The denitrification in the main anoxic reactor during this period was the highest recorded viz. 58 mgNO_3-N/ℓ influent and the high nitrate concentration in this reactor (1.43 mgNO_3-N/ℓ) indicated that this reactor was loaded to it's denitrification potential. The 1.43 mgNO_3-N/ℓ in the main anoxic reactor outflow plus that generated in the aerobic reactor by nitrification of residual ammonia, increased the effluent and sludge underflow nitrate concentrations to over 5 mgNO_3-N/ℓ, which, with the underflow recycle ratio of 1:1, was too high for the pre-anoxic reactor resulting in a nitrate concentration in the pre-anoxic of 0.7 mgNO_3-N/ℓ. The P removal during Sewage Batch 12 was also the highest observed during the investigation, at 13.9 mgP/ℓ. However, because the nitrate load on the main anoxic reactor was too high, this was reduced to 19.25 mgNO_3-N/ℓ influent for Sewage Batch 13. Even though the P removal decreased to 10.25 mgP/ℓ during Sewage Batch 13, the % anoxic P uptake increased to 70% (Fig. 13). The P removal declined because 4.6 mgNO_3-N/ℓ was discharged to the anaerobic reactor which interfered with the BEPR even though the influent RBCOD concentration was high (155 mg/ℓ) and the P release the same as in Sewage Batches 9 to 12.

From Fig. 13, even after completing Sewage Batch 13, a steady state insofar as % anoxic P uptake had not yet been reached. Nevertheless, because the main anoxic reactor denitrification potential had been reached with the nitrate dosing, and hence also the system N removal limit; the influence of the main anoxic reactor nitrate load on the % anoxic P uptake and P removal had been observed; and the P removal appeared to have stabilised even though the % anoxic P uptake appeared to still be increasing, it was decided to report the results obtained so far and make a preliminary comparison of the anoxic uptake BEPR performance with aerobic P uptake BEPR performance (see below).

Sludge settleability

The mean DSVIs observed during the 13 sewage batches are plotted in Fig. 14. The system was started up with a rather poor settling sludge (DSVI ~ 130 mℓ/g) with AA (low F/M) filaments *Microthrix parvicella*, type 0092, type 0041 and type 0675. The DSVI progressively declined and from Sewage Batch 6 the DSVI [*or SVI because when the sludge settles this well, the DSVI and SVI are numerically equal because no dilution is required*] averaged around 50 mℓ/g. The sludge settled very rapidly in the settler and did not form significant sludge blanket in either the internal or final settling tanks. A problem with the sludge was that it did not clarify well and left a fine turbidity in the effluent. This is probably due to the very small aerated mass fraction (0.19). Microscopic analysis of the sludge, in which the filamentous organisms were also identified, indicated very few predatory organisms with the result that the free swimming bacteria were not being grazed as happens in more aerobic systems. Nevertheless, the effluent quality was good and had a low suspended solids (SS) concentration (determined from the difference between the unfiltered and filtered COD concentrations). Only during Sewage Batch 13, were there some suspended solids in the effluent. This was because denitrification was not complete in the main anoxic reactor, resulting in a higher nitrate concentration in the final effluent. This stimulated some denitrification in the final settling tank causing sludge to float to the top of settling tank and escape with the effluent.

The investigation confirmed that system consistently produced a good settling sludge, indeed it stimulated a marked improvement in sludge settleability of the starter sludge. This feature would allow small settling tanks to be built for the system. Preliminary indications are that with such good settling sludges, the two settling tanks required by the system would together not be much larger than the final settling tanks required for the conventional BNRAS system with a sludge of DSVI around 150 mℓ/g, which are quite common. However, this can only be confirmed after a detailed engineering and economic evaluation of the external nitrification BNRAS scheme has been made.

Comparison of BEPR performance

Comparison of measured and calculated P removal

The BEPR performance in the BNRAS system was assessed by comparing the observed P removal with that theoretically calculated from the steady state BEPR model of Wentzel et al. (1990). This model requires as input all the system design parameters (see Table 1) and influent wastewater characteristics including the influent RBCOD concentration. For Sewage Batches 5 to 13, the RBCOD concentration was measured as the difference between the floc-filtered influent and effluent COD concentrations (see Table 2 and 4). The procedure for calculating the theoretical BEPR is given in summary by Ekama and Wentzel (1999) and in detail by Mellin et al. (1998). Basically the procedure "fractionates" the measured VSS mass in the system into active ordinary heterotrophic (OHO) and polyphosphate accumulating (PAO) organisms, OHO and PAO endogenous residue masses and the unbiodegradable particulate material from the influent, viz. $X_{B,H}$, $X_{B,G}$, $X_{E,H}$, $X_{E,G}$ and X_I respectively, by reconciling the theoretically calculated VSS mass with that measured. For this investigation, the influent COD concentration was reduced to 85% of that measured (while keeping the measured RBCOD concentration unchanged) to take account of the 15% COD reduction in the stone column. Knowing the five components of the VSS, the theoretical P removal is matched to that measured by varying the P content of the PAOs ($f_{XBG,P}$) for fixed P contents of the other four VSS components (i.e. at 0.03 mgP/mgVSS).

From the predominantly (>95%) aerobic P uptake BEPR behaviour observed by Siebritz et al. (1983), Wentzel et al. (1985, 1989) and Clayton et al. (1991), the steady state model of Wentzel et al. (1990) has an $f_{XBG,P}$ value of 0.38 mgP/mgPAOAVSS. The $f_{XBG,P}$ values found in this investigation for Sewage Batches 5 to 13, for which the influent RBCOD was measured (see Table 4), are shown in Fig. 15 and ranged between 0.11 and 0.34 mgP/mgPAOAVSS. The mean value for Sewage Batches 5 to 13 is 0.195 mgP/mgPAOAVSS and for the last 5 sewage batches (9 to 13), during which BEPR was good, is 0.235 mgP/mgPAOAVSS. This latter value is about $^2/_3$ of the 0.38 mgP/mgPAOAVSS expected from the aerobic P uptake based BEPR model of Wentzel et al. (1990). However, the effect of the reduced $f_{XBG,P}$ on the overall system P removal is not as much as the ratio 0.23 to 0.38 would indicate because the OHOAVSS, endogenous and inert VSS components, which make up a substantial part of the VSS mass, also contribute to the P removal. Had $f_{XBG,P}$ been equal to the "standard" value of 0.38, then the mean P removal for Sewage Batches 9 to 13 would have been 16.7 mgP/ℓ compared with 12.2 mgP/ℓ observed.

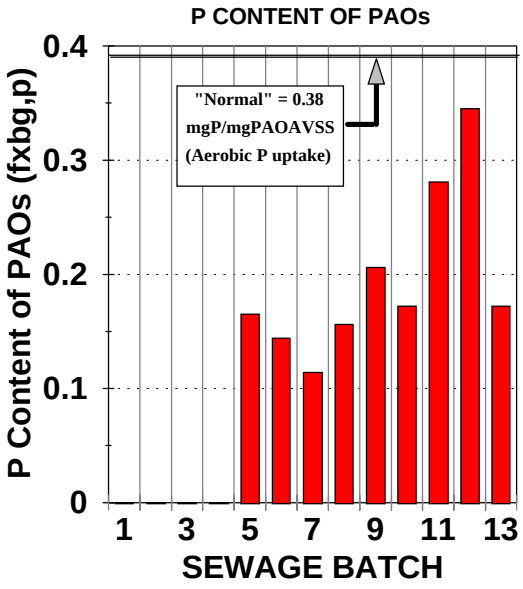

Figure 15
Polyphosphate accumulating organism (PAO) P content ($f_{XBG,P}$, mgP/mgPAOAVSS) to account for the observed biological excess P removal in the BNRAS system, compared with that expected for exclusively aerobic P uptake from the BEPR model of Wentzel et al. (1990).

Different biological P release, uptake and removal behaviour

Anoxic P uptake has also been observed in conventional nitrification denitrification (ND)BEPR systems. It is this type of BEPR behaviour that is sought to be exploited in the DEPHANOX system. However, there seems to be some major differences in P removal performance between aerobic P uptake BEPR and anoxic/aerobic uptake BEPR behaviour. In the UCT and modified UCT systems of Musvoto et al. (1992), Kaschula et al. (1993), Pilson et al. (1995) and Mellin et al. (1998), significant anoxic P uptake (>40%) was observed, which was confirmed with anoxic batch tests on sludge harvested from these systems. In these long-term investigations (>500 d), not only was the excess P removal lower at about $^2/_3$ of that expected from the model of Wentzel et al. (1990), but also the P release to removal ratio was decreased (see Table 5).

In conventional NDBEPR systems, with only aerobic P uptake, the P release/P removal ratio, P removal/influent RBCOD ratio and the P removal/Influent COD ratio are around 3.0, 0.11 and 0.021 respectively (Wentzel et al., 1985, 1989; Clayton et al., 1991, and more recently Sneyders et al., 1997 - see Table 5), and are in conformity with steady state (Wentzel et al., 1990) and dynamic state (Henze et al., 1995) BEPR models. With anoxic/aerobic P uptake these ratios decrease to 1.5 to 2.0, 0.06 to 0.08 and 0.012 to 0.015 respectively and the BEPR is depressed to around $^2/_3$ of that with only aerobic P uptake. It is seems that, given the appropriate conditions, different species of PAOs which accomplish anoxic P uptake find a niche in the system, but which have a significantly lower BEPR performance and use their internally stored PHAs (obtained from anaerobic uptake of VFA which are produced by the OHOs via fermentation of the influent RBCOD) less "efficiently" compared with the aerobic P uptake PAOs.

Comparing the anoxic/aerobic BEPR performance results obtained in this investigation with those observed in the DEPHANOX system by Sorm et al. (1996), it seems that similarly low values are obtained for the P release/P removal, P removal/influent RBCOD and the P removal/influent COD ratios i.e., 0.52, 0.044 and 0.017 respectively with $f_{XBG,P}$ around 0.118 mgP/mgPAOAVSS. However, this comparison should regarded as approximate because the DEPHANOX system of Sorm et al. (1996) showed significant variation in behaviour between it's five "steady state" periods, in particular in the mass of VSS in the system; COD balances couldn't be checked because the aerobic reactor oxygen utilisation rate was not reported; and the influent Total P seemed to have a very low ortho-P fraction making the anaerobic P release estimates spurious.

Modelling anoxic/aerobic P uptake BEPR behaviour

Denitrification by PAOs is included in the biochemical model of Wentzel et al. (1986, 1991) but is not included in the current NDBEPR simulation models (e.g. IAWQ ASM No2 - Henze et al., 1995 and UCTPHO - Wentzel et al., 1992). Therefore anoxic P

TABLE 5
OVERALL COD AND N BALANCE RESULTS AND P RELEASE, UPTAKE AND REMOVAL PERFORMANCE OF THE M/UCT SYSTEMS FROM 5 DIFFERENT INVESTIGATIONS IN THE WATER RESEARCH LABORATORY AT UCT

Parameter	Clayton 1989	Musvoto 1992		Pilson 1995		Mellin 1998	Sneyders	
							CTL	EXP
Temperature (°C)	20	20	20	12	20	30	20	20
N Balance (%)	91	105	98	94	99	82	88	90
COD Balance (%)	92	106	107	84	84	92	92	92
Total P removal (mgP/ℓ)	21.0	12.2	11.3	12.0	10.9	11.4	13.1	16.8
Total P release (mgP/ℓ)	63.0	32.0	32.1	15.0	14.6	19.9	22.8	40.8
% Release in anaerobic	95	64	64	47	15	98	99	95
% Release in 1st anoxic/anoxic	5	36	36	48	56	1	0	5
% Release in settling tank	0	0	0	5	29	1	1	0
Total P uptake (mgP/ℓ)	84.0	44.2	43.4	26.9	26.4	31.0	35.9	57.9
% Uptake in 2nd anoxic/anoxic	5	27	47	47	16	29	0	0
% Uptake in aerobic	95	73	53	53	84	68	99	100
P release/P removal ratio	3.3:1	2.44:1	2.70:1	1.34:1	1.24:1	1.75:1	2.42:1	2.74:1
P rem/infl RBCOD ratio	0.105	0.063	0.060	0.069	0.063	0.082	0.116	0.116[2]
P rem/Total infl COD ratio	0.0210	0.0123	0.0118	0.0121	0.0110	0.0156	0.023	0.024
Remarks[1]	(1)	(2)	(2)	(2)	(2)	(2)	(1)	(1)

[1] Total P removal **does** (1;$f_{XBG,P}$ = 0.38) and **does not** (2; $f_{XBG,P}$ < 0.38) conform to Wentzel et al. (1990) BEPR model.
[2] Used same value as CTL system to calculate the VFA and RBCOD available for BEPR in landfill leachate.

uptake behaviour cannot be simulated with these NDBEPR models. However, proposals to include denitrifying PAOs into the simulation models have been made (Mino et al., 1995; Barker and Dold, 1997; Henze et al., 1998 - IAWQ ASM No2d). This requires resolution of two problems: determination of the concentration of denitrifying PAOs (DPAOs); and reduced P uptake by DPAOs. Barker and Dold (1997) and Henze et al. (1998) deal with the first problem by introducing a factor $\eta_{p,}$ which is the proportion of the PAOs capable of denitrification in the anoxic reactor. This approach is identical in concept to determining the concentration of denitrifying OHOs in the nitrification denitrification (ND) simulation models. With regard to the second problem, Payne (1981) shows that when nitrate/nitrite serve as electron acceptors, ideally only two moles of ATP are formed per pair of electrons transferred to nitrate in the cytochromes instead of three when the transfer is to oxygen (see Casey et al., 1999 for a review of heterotrophic respiratory metabolism). This difference reduces the energy captured when NO_x serves as electron acceptor and hence effectively reduces the yield coefficient under anoxic conditions compared with aerobic conditions. This difference between anoxic and aerobic yield is not included in the steady state (WRC, 1984) and dynamic simulation (IAWQ ASM No1 - Henze et al., 1987; Dold et al. 1991 - UCTOLD) ND models because only a relatively small proportion of the biodegradable COD is utilised under anoxic conditions. In the BEPR metabolic model of Murnleitner et al. (1997, discussed by Brdjanovic et al., 1997), this difference is recognised; under aerobic conditions, the ATP/$NADH_2$ ratio ($\delta_{aerobic}$) is 1.8 mol/mol whereas under anoxic conditions δ_{anoxic} is 0.9 mol/mol, a difference which would account for reduced BEPR with anoxic P uptake. The recently released IAWQ ASM No2d (Henze et al.,1998) does not recognise this different anoxic PAO metabolism, and therefore would not predict a reduced BEPR when significant anoxic P uptake takes place; in fact provided the P uptake process is not rate limited, all the η_p factor does in this model is to allow P uptake to start earlier under anoxic conditions by a part of the PAOs. The model of Barker and Dold (1997) does recognise the different anoxic metabolism; the constant $f_{p,upt}$ is reduced from 0.95 under aerobic conditions to 0.55 under anoxic conditions which reduces the P uptake per PHA metabolised compared with aerobic conditions. However, this model was not validated against experimental systems which exhibited significant anoxic P uptake so that the measure in which this model would reflect the reduced BEPR with significant anoxic P uptake is not known. Because the experiments of Smolders et al. (1995) and Murnleitner et al. (1997) were conducted on systems which were P limited, which may have influenced the BEPR results, in order to gain further insight into the anoxic P uptake behaviour and the mechanisms which stimulate it, kinetic studies on enhanced cultures of denitrifying PAOs under non-P and NO_3 limiting conditions need to be conducted to delineate their kinetics and BEPR capacity.

Conclusions

A biological nutrient removal (BNR) activated sludge (AS) scheme incorporating external nitrification in a fixed media system is proposed. A laboratory-scale evaluation of the scheme indicates that it holds considerable potential for BNRAS system intensification through major reduction in sludge age and oxygen demand and significant improvement in sludge settleability. Because the BNRAS system is not required to nitrify, its anoxic mass fraction can be considerably enlarged at the expense of the aerobic mass fraction creating conditions that allow it to achieve high N removals with domestic wastewaters with high TKN/COD ratios; and promote anoxic P uptake polyphosphate accumulating organisms (PAO) to develop in the system.

From a 250 d laboratory scale evaluation of a 10 d sludge age, 48% anoxic and 19% aerobic mass fraction BNRAS system with

a stone column for external nitrification, it was found that:

- Good COD (~90%) and N (~90%) mass balances were obtained over the system.
- Overall COD removal was excellent at 92% including an approximate 15% COD removal in the stone column.
- Including the nitrate dose to the main anoxic reactor, which increased the effective influent TKN/COD ratio to about 0.14, the system produced a effluent total N (NO_x + TKN) < 10 mgN/ℓ. Nitrification was virtually complete in the stone column and the denitrification in the anoxic reactor was 58 mgN/ℓ.
- The P removal was 10.6 mgP/ℓ and considerable anoxic P uptake took place in the main anoxic reactor. This BEPR is about $^2/_3$ of that expected from the aerobic P uptake BEPR models of Wentzel et al. (1990, 1992) and is probably a consequence of reduced energy capture by the denitrifying PAOs (DPAOs) when nitrate serves as electron acceptor compared with oxygen (Payne, 1981; Murnleitner et al., 1998).
- The oxygen utilisation rate (OUR) was very low - 29 mgO/(ℓ·h) in the 19% aerobic mass fraction. This is about 2.5 times lower than that in conventional nitrification BNRAS systems and arises because the nitrification OUR obtained "free" in the fixed media system and the high denitrification reduces the heterotrophic oxygen demand.
- A very good settling sludge (~60 mℓ/g) was consistently obtained in the BNRAS system. Filamentous organisms *Microthrix parvicella* and types 0092, 0041 and 0675, which are common in conventional BNRAS systems, were identified at low levels in the mixed liquor.

Inclusion of anoxic P uptake PAOs in, and exclusion of nitrifiers from, the biocenosis of the BNRAS system mixed liquor are not essential for achieving BNR in the proposed scheme. However, conditions that promote aerobic P uptake BEPR to maximise this, are also conducive to nitrifier growth. If nitrifiers are supported in the mixed liquor, virtual complete nitrification is essential in the fixed media system to limit nitrification in the main aerobic reactor. If the aerobic reactor nitrate concentration is high, either through nitrification in the aerobic reactor or insufficient denitrification in the anoxic reactor, the pre-anoxic reactor will become overloaded with nitrate resulting in nitrate discharge to the anaerobic reactor and reduced BEPR.

To stimulate anoxic BEPR behaviour, it appears that a high nitrate load and large anoxic mass fraction are required to ensure that the DPAOs are not restricted in their P uptake and denitrification processes; and a small aerobic mass fraction is required to limit aerobic PAO activity. The first requirement needs a high influent TKN/COD ratio. If this is too low (<0.14 mgN/mgCOD for the system design parameters in Table 1) then insufficient nitrate is generated in the fixed media system for the denitrification potential of the anoxic reactor. From this investigation, too low a nitrate load on the anoxic reactor appears detrimental to the development of anoxic P uptake. Reducing the anoxic mass fraction does not resolve this because this increases the aerobic mass fraction which favours aerobic P uptake BEPR behaviour. The TKN/COD ratio of the wastewater can be increased by enhanced primary sedimentation. To stimulate as much BEPR as possible, and counter the reduced BEPR with anoxic P uptake, the primary sludge can be acid fermented to generate additional volatile fatty acids (VFA) and readily biodegradable (RB)COD. This approach (see Fig. 2) will reduce overall COD load on the system (and hence reduce the system volume and oxygen demand), and increase the TKN/COD ratio and VFA/RBCOD concentration, which would stimulate more DPAOs growth in the system at the expense of the OHOs. However, before the external nitrification BNRAS scheme can be implemented at large scale, an engineering and economic evaluation is required to quantify its potential benefits and savings.

Acknowledgements

This research was supported jointly by the Water Research Commission, Water and Sanitation Services, a South African subsidiary of Lyonnaise des Eaux, and the Foundation for Research Development and is published with their permission.

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